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Land use impact assessment of margarine

Llorenç Milà i Canals · Giles Rigarlsford · Sarah Sim

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Abstract
Purpose This paper presents a case study of margarine, demonstrating the application of new characterisation factors (CF) for land use and a number of land use change impacts relating to biodiversity and ecosystem services. The objectives of this study were to generate insights as to the ease of applying these new factors and to assess their value in describing a number of environmental impacts from land use and land use change relating to the margarine product system.
Methods This case study is a partial descriptive life cycle assessment (LCA) of margarine. The functional unit of the study is 500 g of packaged margarine used as a spread in the UK and Germany. The life cycle stages included were: agricultural production, oil processing, margarine manufacture and transportation to regional European distribution centres. Essential for the application of the new CF was the identification and quantification of the inventory flows for land occupation (land use) and land transformation (land use change) flows. A variety of methods have been applied to determine the inventory flows for the agricultural and industrial stages in the life cycle. These flows were then assessed using the new CF and land use-related environmental impact categories recommended in this special issue.
Results and discussion Land occupation was the major determinant for all the new impact categories with the exception of the water purification potential. Many of the impact categories followed a similar pattern and therefore, the inventory result for land occupation in this case study explains a large share of most of the impacts. Where land occupation alone is not a suitable proxy for environmental impacts (i.e. for freshwater regulation potential), differentiation at the level of biomes has proven relevant. In addition, the land use types distinguished so far were found to be useful in highlighting likely hotspots in the life cycle, although further differentiation of 'agricultural land' is suggested to account for the differences between annual and permanent crops.
Conclusions The new land use impact assessment methods applied help to identify hotspots in the life cycle of margarines, with different proportions and sources of vegetable oils. The specific impacts of each vegetable oil are determined mainly by the yield (and thus land occupation), but also by the type of agriculture (annual vs. permanent crops) and the sourcing location (and thus the sensitivity of biomes and occurrence of land use change). More research is needed to understand the usefulness of the various impact categories. For land use types, further refinement is required to describe different agricultural systems consistently across impact categories (e.g. annual vs. permanent cropping). In addition, the conceptual basis for the CFs applied in this case study (i.e. use of a potential reference for occupation and transformation) has limitations for some types of decisions normally supported by LCA.

Keywords Biodiversity · Ecosystem services · Land use impacts · LCIA · Life cycle impact assessment · Margarine · Vegetable oils

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L. Milà i Canals (✉) · G. Rigarlsford · S. Sim
Safety and Environmental Assurance Centre, Unilever R&D, Colworth Science Park,
Sharnbrook, Bedfordshire MK44 1LQ, UK
e-mail: Llorenc.Mila-i-Canals@Unilever.com

1 Introduction

Work conducted under the UNEP/SETAC Life Cycle initiative to derive characterization factors (CF) for land use (LU, also called occupation) and land use change (LUC, also called transformation) impacts on biodiversity and ecosystem services (Koellner et al. 2012a) has recently been completed; this paper assesses the application of these new CFs to LU- and LUC-related impact categories in a case study of margarine. The specific goals of the case study are to:

1. Understand the additional value of applying the new CF in terms of describing and estimating the size of the environmental impacts associated with margarine
2. Understand the ease of use of the new CF including the identification of any difficulties in adapting current inventories or finding additional life cycle inventory (LCI) information to facilitate the use of the CF (e.g. information on spatial differentiation)

Over the years, a number of studies by Unilever have been conducted to understand the environmental impacts of margarine to inform supply chain management. Two studies have been published externally (Shonfield and Dumelin 2005; Nilsson et al. 2010) though neither study assesses the environmental impact of the margarine product system on biodiversity nor ecosystems services since no suitable methodologies were available. However, since margarine is a product principally from land-based agricultural systems (made predominantly from vegetable oils), environmental impacts relating to biodiversity and ecosystem services are highly relevant. This case study uses as its basis the most recently published study (Nilsson et al. 2010) on margarine.

2 Methodology

The methodological approach in this case study is a descriptive (attributional) life cycle assessment (LCA; Nilsson et al. 2010). The functional unit of the study is 500 g of packaged margarine used as a spread. The life cycle stages included were: production of raw materials (ingredients and packaging),[1] oil processing, margarine manufacture and transportation to regional distribution centres and disposal of consumer packaging materials and packaging materials used in transportation. Onward transport to retail units (e.g. shops/supermarkets) and to the consumer's home, and storage in the retail units and at home were excluded. Whilst Nilsson et al. (2010) considered margarine sold in the UK, Germany and France, here we consider just the UK and Germany since the products studied in these countries represent the two extremes in terms of land use aspects namely, fat content (the UK margarine of 38% versus German of 70% fat) and different profiles in terms of the type and sources of vegetable oils (see Table 2 in Nilsson et al. 2010 and Fig. 3 in this paper).

This paper focuses on the approaches to assess the environmental impacts related to land use and land use change. Two main issues merit attention here, both pertaining to the LCI phase of the study. Firstly, the identification and quantification of land occupation flows linked to the new classification system (Koellner et al. 2012b), which is explained in Section 2.1. Secondly, the quantification of land transformation flows in the relevant countries for the crops, which is explained in Section 2.2. In addition to the description of land use flows in the foreground system, the adaptation of secondary datasets in existing databases to be used as proxies is explained in Section 2.3.

The land occupation and transformation flows identified have then been characterised with the CF recommended in this special issue, as follows:

- For the biodiversity damage potential (BDP), the approach and CF offered by de Baan et al. (2012) were used. Average world CF were used for those biomes not covered in de Baan et al. (2012). In addition, de Baan et al. (2012) do not provide CF for transformation flows; therefore CF were calculated for this case study and they are reported in the Electronic supplementary material (ESM) (1).
- For climate regulation potential (CRP), the approach suggested by Müller-Wenk and Brandão (2010) was used. An average CRP for forest and grassland was used to represent plantations, and the transformation factors from human land uses (e.g. agriculture) were considered with the inverse sign as those for the transformation to those land uses; this assumption is not fully consistent with Müller-Wenk and Brandão (2010) but the amounts of transformation to natural land are insignificant and thus do not affect the results.
- For biotic production potential (BPP), the approach and CF offered by Brandão and Milà i Canals (2012) were used.
- For impacts on Ecosystem Services, other than CRP and BPP, the approach and CF proposed by Saad and Margni (2012) were used. These include impacts on: freshwater regulation potential (FWRP), erosion regulation potential (ERP), and water purification potential (WPP) assessed here by two indicators related to physico-chemical filtration (WPP-PCF) and mechanical filtration (WPP-MF).

A compilation of the CF flows used in this case study, with cross-references to the relevant publications, is provided in ESM (1).

[1] A cut-off criterion was applied so that ingredients constituting less than 0.5% of the weight of the total ingredients were omitted.

2.1 Identification and quantification of land occupation flows

2.1.1 Land occupation in agricultural stages

Nilsson et al. (2010) quantify the total amount of land occupation in the agricultural stages for both margarine products (see Table 8 in Nilsson et al. 2010). In this paper, the amount of land occupied was updated for linseed, maize and cassava using Food and Agriculture Organization (FAO) data and land occupation in subsequent supply chain stages is also included (Section 2.1.2). In addition, the agricultural land uses were ascribed to the most likely country of crop production (according to Unilever supply managers). This was done to allow spatial differentiation in the life cycle impact assessment (LCIA) phase, as recommended by Milà i Canals et al. (2007) and Koellner et al. (2012a). The ingredients used in the two margarines and their source country are specified in Table 1; the expected biome within each country was determined by expert judgement. For land use flows in the background system (e.g. for energy delivery, fertiliser production, etc.), no further spatial differentiation has been provided, and generic "global" characterisation factors have been used in the impact assessment phase.

All crops were assumed to require the use of land for 1 whole year, even though many are actually in cultivation for less than this. The rationale for this is as follows: sunflower is harvested late in the year and therefore nothing else is sown that year; following the harvest of rapeseed, green manure is grown and this is not considered to be a harvested crop, but rather a soil-improvement measure; oil palm fruit is a permanent crop; linseed is grown in Canada where winters are very cold and therefore nothing else is grown; the ground is left fallow following the cultivation of maize; and cassava is grown and harvested over a number of years. Therefore, land occupation (hectare-year) per kilogram for these crops is obtained as the inverse of the yield in Table 1. The yields considered for the main ingredients are considered representative for the main suppliers; however, large variability would be expected if FAO data had been used instead. The effect of yields on the results was tested for the main crops.

2.1.2 Land occupation in post-agricultural stages

Land occupation was also considered for oil mills, oil refining and margarine production. The land area for all of the oil mills was estimated as follows based on available information for palm oil mills in Malaysia. The amount of concrete (2,850 t mill and boiler house) used for foundations is provided in the inventory for capital goods in a recent study (MPOB 2010, chapter 4). The area of hard standing/floor was then estimated using a factor for the amount of concrete per square metre (740 kg/m^2) taken from Jönsson et al. (1998). The land occupation of the actual building was calculated by dividing the area of factory floor by the amount of palm fresh fruit bunches (FFB) processed each year (270,000 t/year) giving 0.014 m$^2\times$year of industrial land occupation per tonne FFB processed. In addition to the land use by the actual building, 0.041 m$^2\times$year per tonne of oil was considered as occupation of urban green areas for the mills; this refers to paths and vegetated areas between buildings, and is based on the ratio of industrial area and urban green areas given for oil refineries and margarine factories below. These figures were also used for all the other oil crops.

An occupation of 0.26 m$^2\times$year as industrial area per tonne production and 0.73 m$^2\times$year of urban green areas per tonne for each of the oil refineries and margarine factories was based on Unilever's German margarine manufacturing site. This could be a slight overestimate as in some cases refining activities take place in the margarine factory.

Table 1 Main country of origin, related biomes, and yields (kilogram harvested produce at farm gate) considered in this study for the vegetable oils used in the margarines

Crop	Source country	Biome	Crop Yield (kg[a] ha$^{-1}\times$year^{-1})
Sunflower seed	Argentina	Temperate grasslands, savannas and shrublands	1,500[b]
Sunflower seed	Russian Federation	Boreal forests/taiga and Temperate broadleaf and mixed forest	1,500[b]
Sunflower seed	Ukraine	Temperate broadleaf and mixed forest	1,500[b]
Rapeseed	Germany	Temperate broadleaf and mixed forest	4,250[b]
Oil Palm Fruit	Malaysia	Tropical and subtropical moist broad-leafed forest	25,000[b]
Linseed	Canada	Boreal forests/taiga	1,338[a]
Maize (seed)	Germany	Temperate broadleaf and mixed forest	9,250[a]
Cassava (tapioca, tuber)	Thailand	Tropical and subtropical moist broad-leafed forest	21,025[a]

[a] FAOSTAT (2011a)

[b] Nilsson et al. (2010)

2.2 Quantification of land transformation flows

2.2.1 Land transformation linked to agricultural stages

As shown in Fig. 1, a three-step approach was used to determine whether a crop grown in a specific country was potentially related to any land transformation (LUC), and what transformations were involved if any. A 20-year time period was considered as this is often recommended for the allocation of impacts of land use change (see e.g. Koellner et al. 2012a; Flynn et al. 2011). Because in this case the average LUC in the whole country rather than a specific plantation was assessed, there was no need for allocating it to the first 20 years of land use (as suggested in BSI 2008; Koellner et al. 2012a; Flynn et al. 2011). In order to smooth out short-term fluctuations in land use, 5-year averages were used. FAO Statistical Database (FAOSTAT 2011b) data were used to perform this analysis.

For example, the average area harvested for oil palm in Malaysia grew from 1,925,437 ha (average between 1990 and 1994) to 3,654,600 ha (2004–2008; FAOSTAT 2011b). In step 2 (Table 2), it was determined that permanent crops in Malaysia (oil palm is a permanent crop) also grew during the same period, by a total of 251,600 ha to an average of 5,785,000 ha in 2004–2008, i.e. in step 3, a total land transformation of 251,600/5,785,000=0.0435 ha (435 m^2) of LUC for every hectare of land being used every year. The land use types that decreased in area between 1990 and 1994 and 2004–2008 are arable land (by −16,200 ha) and, most notably, forest area (by −1,405,120 ha, or 98.9% of the total LUC). In summary, for every hectare-year used for permanent crops (and thus oil palm) in Malaysia in 2004–2008, there were 430 m^2 of forest and 5 m^2 of arable land converted to plantation. Note that the time periods used for the LUC calculations were shorter than 20 years for all countries due to data availability. The data used to assess LUC in all other crops and countries are included in ESM (1).

The approach outlined here differs from the one taken in Nilsson et al. (2010) in which a 'worst case' scenario for LUC associated with palm oil was adopted for the sensitivity analysis (in that case focused on estimating greenhouse gas (GHG) emissions associated with LUC) based on recommendations described in PAS2050 (BSI 2008). In that scenario, it was assumed that 100% of palm oil was sourced from land which had undergone transformation from forest in the last 20 years, namely 50% in Malaysia and 50% in Indonesia. LUC estimates were not made for any of the other vegetable oils used in the margarine products studied. It was considered in this study that the worse case scenario for palm oil was not representative as not all purchased palm oil is sourced from land transformed in the last 20 years. Therefore in this paper, we have refined the approach for estimating LUC so that it is more realistic and also allows for spatial differentiation (required to apply the CFs). Note

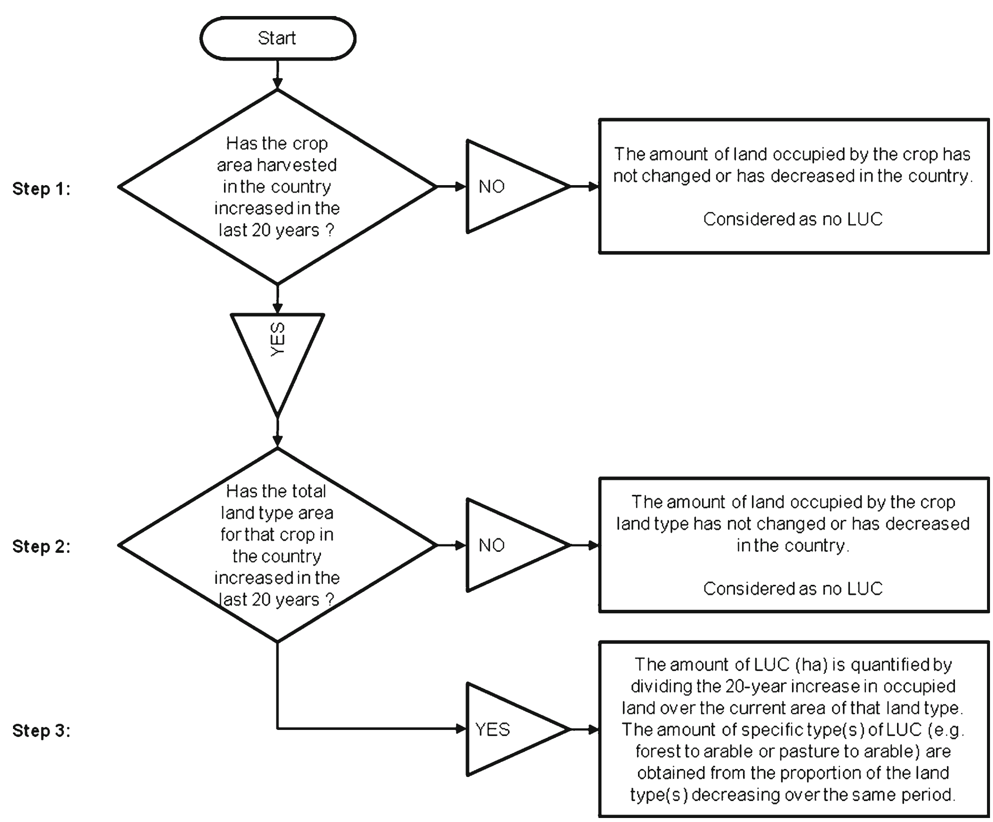


Fig. 1 Decision tree to determine the existence and magnitude of land use change for the studied crops in the sourcing countries

Table 2 Land use changes between arable crops; permanent crops; and forest in Malaysia for the periods, 1990–94 and 2004–08

	Average area for years 1990–1994 (1,000 ha)	Average area for years 2004–2008 (1,000 ha)	Change over analysed period (1,000 ha)	Land use change allocated to each year of land use (ha×year)
Arable land, Malaysia	1,816	1,800	−16	−0.0005
Permanent crop, Malaysia	5,533	5,785	252	0.0435
Forest area, Malaysia	22,219	20,814	−1,405	−0.0430

Source: FAOSTAT (2011c) and own calculations

however that our refined approach identifies net changes in LU type and thus ignores any LUC occurring within land use types that does not lead to a change in total area devoted to each type. As a result, e.g. shifting cultivation would not be captured by the statistics unless total area changed, even though by definition such agriculture is always linked to LUC. Note also that when applied at a country level, this approach does not detect indirect LUC, caused by displacement of crops to be grown in other countries. This is a limitation of this attributional-focused approach; see, e.g. Kløverpris et al. (2007) or Brandão (2011) for approaches to deal with indirect LUC.

2.2.2 Land transformation linked to crop processing

LUC was only considered for the palm oil mill and not for the other industrial sites, considering the same fraction of LUC as for oil palm plantations (435 m^2 per ha-year). The rationale for this was that palm oil mills are normally situated close to the palm plantations and significant LUC change was only determined for palm oil and not the other oil crops.

2.3 Modelling land occupation and transformation in secondary datasets

The life cycles of the margarine products sold in Germany and the UK were modelled in commercial LCA software (GaBi 4 from PE International). Both primary data (i.e. cultivation of palm fruit, rapeseed and sunflower) from specific studies on oils (Shonfield and Dumelin 2005) and secondary data (i.e. maize, linseed and cassava[2]) obtained from the ecoinvent database were included in the model. Secondary oil crop inventory data were modified to better represent the land use flows (both occupation and transformation) in the relevant sourcing countries (see Table 1). In order to incorporate land occupation into the secondary LCI datasets for crops as per the FAOSTAT yields (Section 2.1.1), the existing land occupation flows in the relevant ecoinvent processes were altered. The actual land occupation in the agricultural stage was identified from the ‘unit process-single operation’ (u-so) and this enabled the ‘aggregated-LCI result’ (agg) to be corrected as illustrated below for maize (Table 3). The same type of adjustment for LU flows was applied to the secondary LCI data sets for oil processing mills (Section 2.1.2).

A similar correction was required for land transformation/land use change associated with maize, but not for the other secondary data sets used for oils as these were not related to crops for which land transformation was identified.

The majority of secondary datasets used in background processes in the rest of the margarine’s life cycle (e.g. transportation, packaging production, energy delivery, etc.) were taken directly from ecoinvent without further changes in the occupation and transformation flows reported.

3 Results

3.1 Impact assessment of margarines

Figure 2a–g provide the contributions per functional unit of the different ingredients and life cycle stages of the two margarines for the seven land use-related impact categories. For most impact categories (except FWRP), the UK margarine with 38% fat content shows larger total impacts than the German margarine with the higher 70% fat content. This is because the UK margarine contains a higher proportion of oils from low-yielding crops such as sunflower (yield, 1.5 t/ha). Figure 3 provides a graphical representation of the composition of the two margarines in order to facilitate the interpretation of the results. Sunflower growing dominates the impact results for the UK margarine (where it represents about 25% of the ingredients) and it also has a significant contribution towards the impacts of the German margarine in which it is only 3.5% of the ingredients. In comparison, the impacts from rapeseed are generally lower even though it represents 36% of the German margarine. This can be explained by the higher yield for rapeseed (4.2 t/ha) compared to sunflower. Palm oil, which makes up ca. 26% of the German recipe, has a relatively low contribution to all the land use impact categories, even though significant LUC of 435 m^2/ha×year of tropical

[2] Proxy datasets were used for linseed and cassava as described in Nilsson et al. (2010).

Table 3 Adjustment of land occupation flows in ecoinvent agricultural processes: Switzerland: maize integrated production (IP), at farm

Flows	u-so ($m^2 \times year$)	agg ($m^2 \times year$)	updated agg ($m^2 \times year$)
Land occupation, arable, non-irrigated (Hemerobie ecoinvent)	0.5388	1.3217	0.7829
Land occupation, agriculture, arable, intensive, temperate broadleaf and mixed forests [land use/occupation (LU)]	–	–	1.0810

u-so unit process-single operation, *agg* aggregated-LCI result

ecosystems with high biodiversity and carbon values was allocated to this crop. This can be explained by the lower impacts associated to permanent agriculture for most impact categories relative to annual crops.

In conclusion, the type of oil, not only the total fat content, determines the overall impact and this is mainly due to the differing yields of the various oil crops and hence the land occupation level. In addition, the production system of the various oil crops (annual vs. permanent crops) and the sourcing region (biome) have a significant effect on the contributions to the impact categories, as explained below.

3.2 Specific drivers for the impact categories

The impacts on BDP and CRP (see Fig. 2a–b) are the largest for the UK margarine because of the contribution from low oil-yielding crops (sunflower, linseed). When FAO derived country-specific average, yields are considered (instead of the constant yield across countries, see Table 1), the results for some of the impact categories varied by as much as 5–8%, with the UK margarine still showing higher impacts. This is in spite of the fact that the total fat amount of these two oils in the UK recipe is about half the amount of rapeseed oil and palm oil in the German recipe. The ERP (Fig. 2c) follows a similar pattern to BDP and CRP but the contribution from palm oil is almost negligible[3] and the transformation to arable land linked to rapeseed in Germany has a very significant contribution (about 20% of the total).

In the case of impacts on freshwater regulation (FWRP, Fig. 2d), the German margarine has a higher impact than the UK product. This is because a large percentage of the UK ingredients (linseed from Canada and sunflower from Argentina Russia and Ukraine) are sourced from different biomes rather than from within a single biome (e.g. '04 Temperate broadleaf forests'). This is notable also because from an LCI perspective, the German margarine has the lowest land occupation overall of the two margarines (1.51 versus 2.36 m^2 $year^{-1}$ for UK). Thus, the FWRP is the only impact category in this study where spatial differentiation at the biome level is important. Note that BDP is measured with a relative indicator of the proportion of species affected; if an absolute indicator was used, then the differentiation of biodiversity-rich biomes or those with populations of species with high sensitivity to loss (tropical forests) would probably be more prominent in the results.

The WPP impact categories (WPP-MF and WPP-PCF, Fig. 2e–f) deserve some special attention because they indicate a very significant contribution from land transformation flows linked to non-agricultural or 'sealed' land use (e.g. industrial buildings; transport infrastructure) associated with the packaging's life cycle, product manufacturing and road distribution. This is because the CF for land transformation to sealed land flows are 3–5 orders of magnitude higher than for land transformation to agricultural land (Saad and Margni 2012).

Finally, the impact profile for biotic production (BPP, Fig. 2g) follows a very similar pattern to the WPP impact profile, but with smaller contributions from palm oil due to the same soil organic carbon being considered for forests and permanent crops (Brandão and Milà i Canals 2012). In BPP, there are also relevant contributions associated with the packaging component of the product due to the occupation of sealed land rather than transformation flows as in the case of the two WPP impacts (see above).

3.3 Relative contributions from land occupation and land transformation flows

As discussed above, the biodiversity and ecosystem services impacts are largely driven by land occupation as opposed to land transformation for most impact categories (see Fig. 4a–d and g) with the notable exception of water purification potential (both mechanical and physico-chemical, see Fig. 4e–f). Figure 5 provides the contributions to the LCI occupation flows (m^2year) and they are clearly dominated by the agricultural stage. It could therefore be argued that the inventory indicator 'land use' (called land occupation, or competition, in LCA, see, e.g. Lindeijer et al. 2002; Milà i Canals et al. 2007; Nilsson et al. 2010) is a good initial proxy for many of the land use-related impact categories applied in this study and product system. However, land use or occupation is not a good proxy for impacts when the CF vary significantly

[3] This is due to the fact that the CF for tropical forests was used for oil palm plantations.

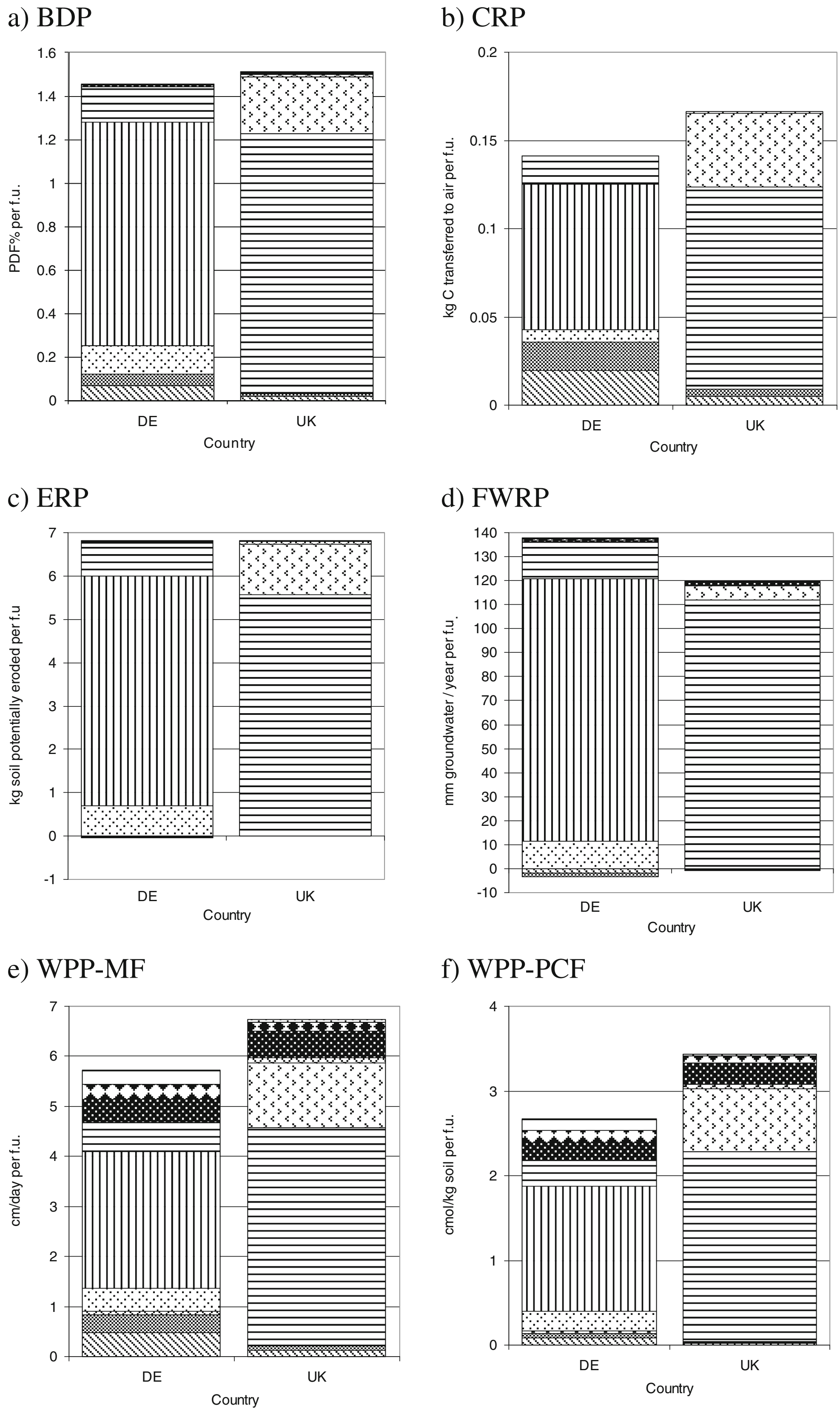
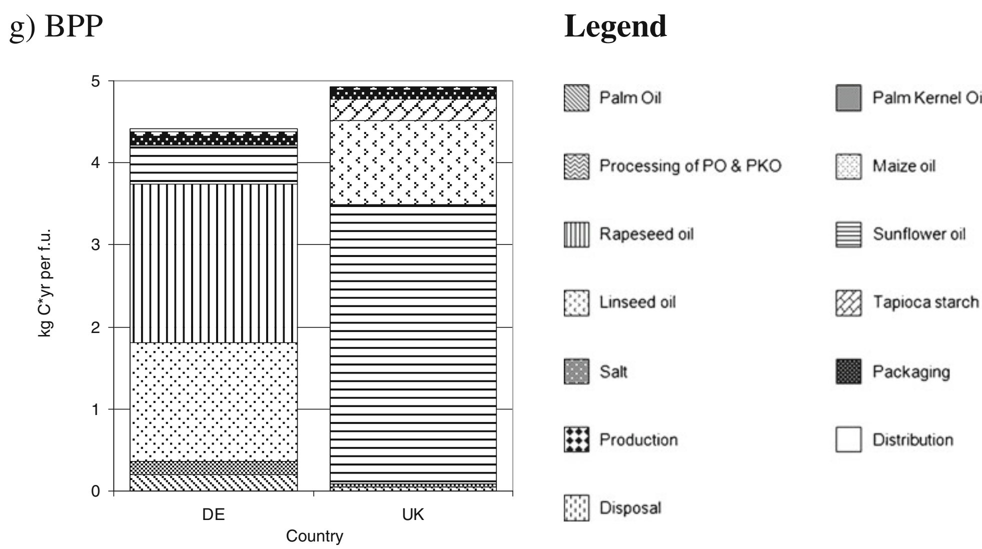


Fig. 2 Contributions of the different ingredients and life cycle stages to the land use impact categories: **a** *BDP* biodiversity damage potential, **b** *CRP* climate regulation potential, **c** *ERP* erosion regulation potential, **d** *FWRP* freshwater regulation potential, **e** *WPP-MF* water purification potential through mechanical filtration, **f** *WPP-PCF* water purification potential through physico-chemical filtration, **g** *BPP* biotic production potential. All impacts expressed per functional unit (f.u.): 500 g tub of margarine to be used as spread

depending on bio-geographical differentiation (i.e. for FWRP, Fig. 2d) or when the CF vary by orders of magnitude between different land use flows (occupation or transformation; e.g. for WPP and BPP, Fig. 2e–g).

Fig. 2 (continued)



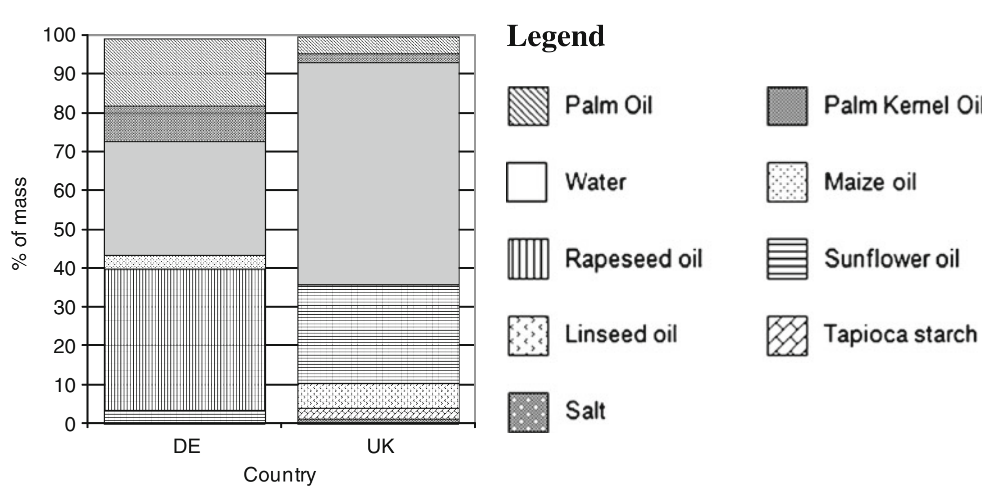
4 Discussion

Until recently, assessment of land use impacts in LCA was not included or limited to one or two impact categories, usually biodiversity or soil quality (see e.g. Brandão et al. 2011). Such impact categories were often assessed in a spatially generic way or with CF adapted to the main location of production only (see e.g. Schmidt 2008). The land use impact assessment methods provided in this special issue by the UNEP/SETAC Life Cycle Initiative project enable the assessment of land use impacts in LCA on a global scale for the first time. Differentiation and coverage has been improved for a number of impact categories covering biodiversity and some ecosystem services and the main land use types. Furthermore, the land use impacts provide for global geographic coverage at the level of the biomes where the impacts occur. The first point for discussion is whether the level and extent of differentiation is sufficient in these three areas (impact categories; land use types; geography). In terms of the seven new impact categories, the results of this case study suggest that there is a significant overlap between some of the impact categories, and most of them are actually determined to a large extent by land occupation. However, it is proposed that this is further investigated with case studies involving a diversity of land uses in different biomes.

As described in Sections 1 and 2, this study took the same scope and system boundaries for margarine as those studied by Nilsson et al. (2010). In this study, land occupation was found to be the key driver of the new impact categories. Therefore, the conclusions of the Nilsson study that margarine has smaller environmental impacts than butter remains valid since butter production required double the land occupation of margarine.

The land use types distinguished so far (mainly at the first level of classification as suggested by Koellner et al. (2012b)) are useful in highlighting the likely hotspots in the life cycle. In this case study based on a food product, most impacts were dominated by the agricultural stages but the importance of non-agricultural or ‘sealed’ land uses was shown for some impact categories. A clear need identified

Fig. 3 Composition of the two studied margarines



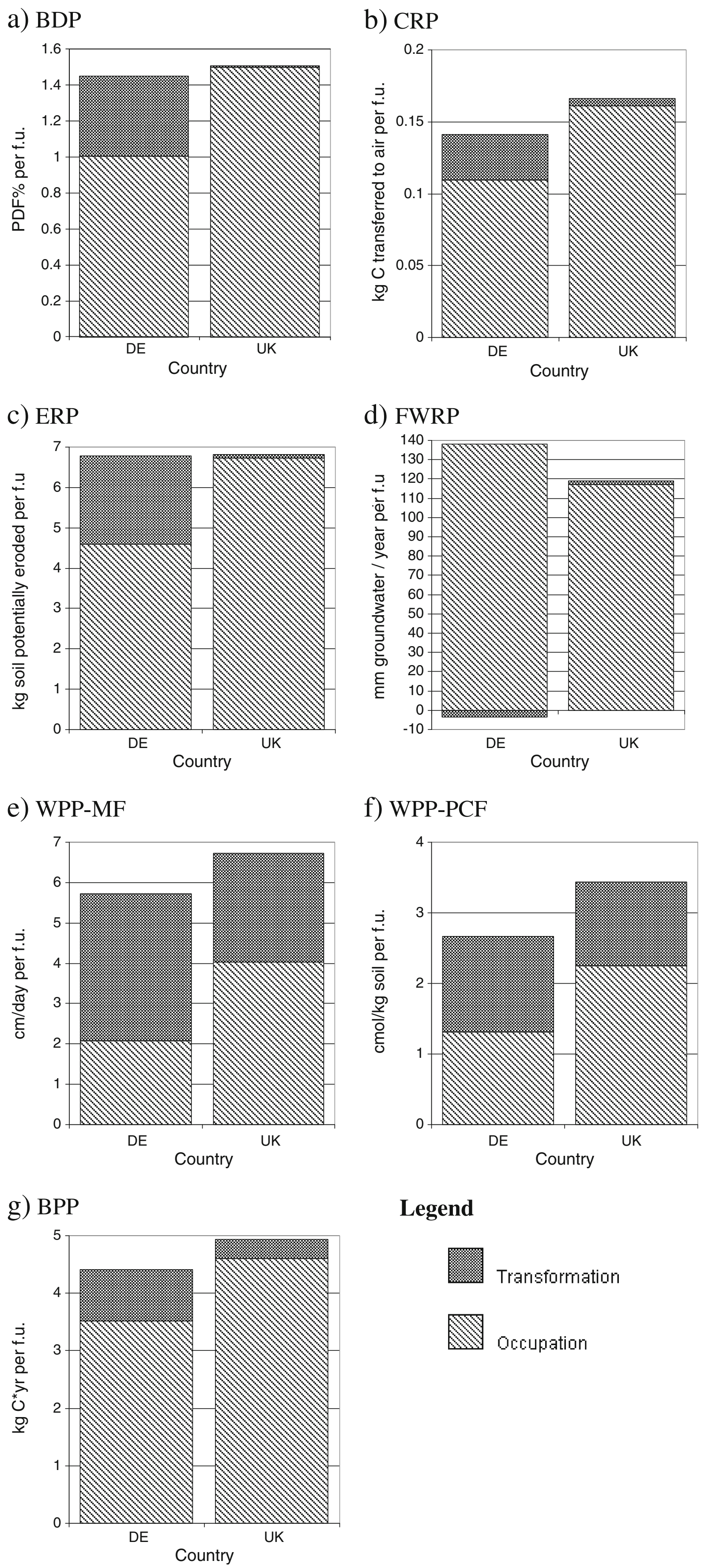


Fig. 4 Contribution of land occupation and land transformation flows to the land use impact categories (see Fig. 2 for their description). All impacts are expressed per functional unit (f.u.): 500 g tub of margarine to be used as spread in Germany and the UK

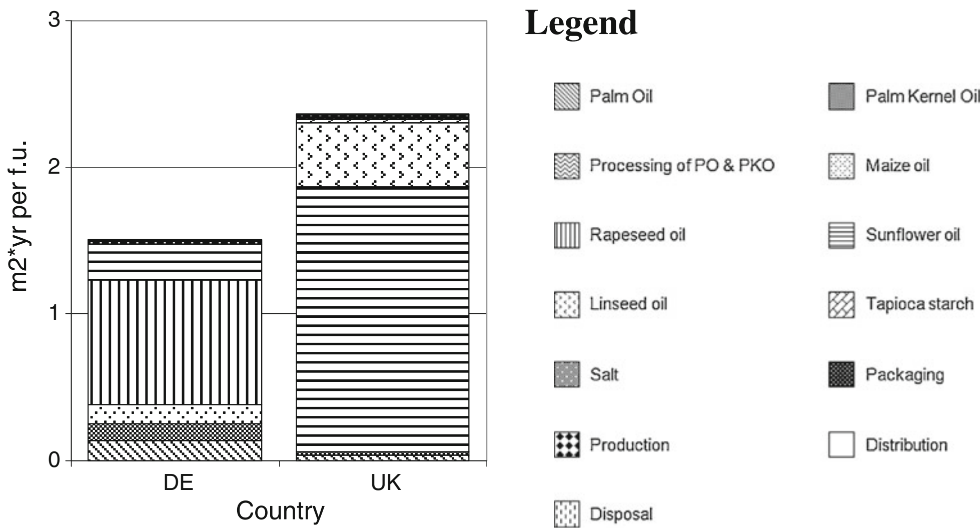


Fig. 5 Contribution to land occupation flows by the different ingredients and life cycle stages (square meter per year per functional unit)

for further refinement is in the types of agricultural production. Perennial crops, such as oil palm plantations, are likely to have significantly different impacts on several impact categories when compared to annual arable crops. Therefore, it is not sufficient to provide CF only for "agricultural land" at the first level of land use classification. In this paper, CF for forest were used for some impact categories (ERP, FWRP, WPP)[4] to represent oil palm plantations, as they were believed to be more appropriate. This assumption needs to be reviewed when more refined CF are provided that allow for a distinction between perennial and annual crops. In addition, given the importance of occupation in the overall impacts, further refinement in the land use classification with regards to agricultural management would be desirable (e.g. different tillage regimes, organic vs. conventional agriculture, etc.). However, assigning information on such management at a large scale, or when crops are purchased as commodities in the open market, will remain a challenge. Crop rotations also play an important role, e.g. in preserving soil quality, even though most annual crops are grown in a rotation, the effects of such rotations have not been assessed in this study. Such effects cannot really be taken into account either in generic CF for 'agriculture, arable'; however, where the effects of different crop rotations are part of the studied system such effects may be included in the impact assessment phase with specific CF calculated by the practitioner.

Where land occupation alone may not be an adequate proxy for environmental impacts (i.e. for FWRP), differentiation at the level of biomes has shown to be relevant in this case study. It remains to be studied whether finer levels of bio-geographical differentiation would provide more informational value to such studies but this would need to be balanced against its practical feasibility as discussed below.

[4] In addition, for CRP, an average value between pasture and forest was used, as this was considered a better fit for plantations.

Land transformation (LUC) flows do not have a very significant effect on the impact results, except for those impact categories where CF for transformation flows are significantly larger than for the corresponding land occupation flows. This is due to a combination of factors including: (1) land transformation and occupation impacts are calculated in relation to the same reference (i.e. regeneration to an ideal potential quality/natural climax steady state) and (2) land occupation flows are bigger because land transformation does not always occur, and when it does it is allocated over 20 years of land use or the current total land used in each country as calculated in this study. In addition, CF for occupation and transformation flows are not often orders of magnitude different because the modelling period for their impacts is limited to 500 years and the indicators used can only vary within a limited range (e.g. BDP is assessed as relative amount of species). Consequently, land occupation tends to dominate the impact results unless very long regeneration times are considered in the calculation of transformation CF. Such a modelling approach is actually contrary to current opinion and policy in the area of land use which attributes many land use related issues (e.g. GHG emissions and biodiversity loss) to land use change. This limitation of the UNEP/SETAC LCI framework for modelling impacts from land use change is related to the assumption of maintaining the land quality in terms of an idealistic potential quality which may never be reached again in reality. In this sense, the results of the impact assessment need to be interpreted as a view of the differences in biodiversity or ecosystem services that are being maintained with respect to an ideal or theoretical potential rather than a description of the actual change in land quality. The type of decisions that may be supported with such impact assessment are those considering long-term effects of hypothetical land use policies such as the consequences of allowing forest re-growth in temperate regions. However, if decision makers wish to

consider shorter-term effects of land use change, the modelling framework and associated impact categories could actually lead to the wrong decision. For example, if we deforest the tropics to grow more palm oil, we will still have more actual (present) emissions of GHG even if the potential difference between the amount of carbon sequestered (and thus the impact on CRP) in the plantation and the tropical forest is smaller than that between a rapeseed crop and a temperate forest (as suggested in this paper). The 'foregone sequestration' represented by CRP (and other impact potential assessed in this paper) is important for all crops, not only for palm oil. However, direct emissions from deforestation may dominate greenhouse gas balances of tropical crops because deforestation is occurring now. In the CRP, we look at the loss in carbon sequestration, i.e. also considering that there was LUC a long time ago in Europe. For this reason, the conceptual basis for the CF applied in this case study (and in particular the reference used to derive the CF) only partially informs some types of decisions normally supported by LCA—e.g. the optimisation of individual product systems or the management of brand/company portfolios with respect to their current impacts. The results of the land use impact assessment using this type of framework would, therefore, have to be used together with other commonly used LCA impact categories. In this context, the new land use impact categories inform of potential impacts on biodiversity and ecosystem services as well as the 'opportunity cost' impacts of not letting land regenerate.

In terms of the ease of using the new characterisation factors, this study illustrates a number of challenges. These are related both to (1) the existing data 'infrastructure' in LCI databases and (2) data gathering challenges in product supply chains. When secondary data inventories (ecoinvent) were used in the foreground, the existing LU and LUC flows were edited, as described in Section 2.3. This had to be performed manually and involved time-consuming data tracking in order to identify which land use flows were related to those parts of the system that needed updating. In addition, the procedure followed in ecoinvent 2.0 to quantify transformation flows is not always in line with the common definition of land transformation (e.g. shifting from non-irrigated agriculture to irrigated agriculture is recorded as a land transformation flow in ecoinvent, when it is only a change in management). No attempt was made to provide spatial differentiation information to the background data because at the level at which such data are aggregated it would have been impossible. This illustrates the need for greater consistency or standardisation in how LU/LUC flows are considered in LCI databases.

The results of this case study showed that the impact of LUC in the foreground system is smaller than the LUC in the background system (e.g. packaging, transport, etc.). This highlights differences in the calculation, including allocation, of LUC between Ecoinvent and in this case study (explained in Section 2.2.). This reduces consistency in the inventory phase and thus in the LCA results, and therefore greater consensus on the calculation of the amount of LUC is desirable. However, inconsistencies generated by the different LUC estimation methods are unlikely to be larger than the inconsistencies generated by having physical and economic allocation methods mixed in LCI databases. In the case of the ecoinvent database, land transformation flows in agriculture do not have an effect on the impact assessment phase as the 'from' and 'to' flows are set to cancel each other out when there is no actual transformation. However, the contribution from ecoinvent datasets to the LCI indicator for land transformation is misleading because all hectares of land transformation ('to' and 'from') contribute to this indicator. In this paper, we have suggested a calculation procedure to estimate direct land use change from FAO statistics, which is consistent and easy to use for any crop in the world. As noted above, though, this approach ignores indirect LUC caused by displaced production of crops outside the country being studied.

Given the localised/regionalised nature of impacts on biodiversity and ecosystem services, the CF for the new impact categories have been developed to allow for spatial differentiation. However, currently it is not always the case that the requisite inventory data are available. For margarine, this is exemplified by the fact that vegetable oils are often traded as commodity products. This means that the origin of the oils may not always be known, particularly at a country level. This is simplified by working at a biome level rather than at a country level but only if the most representative biomes for each type of crop are known. Additionally, the sourcing of some food products may be seasonal or dependent on a range of factors such as cost and continuity of supply and therefore it may be common that some non-commodity ingredients will also be sourced from several countries. In such cases, it may only be possible to obtain estimates of quantities from the source countries on an annual basis. This illustrates a potential disparity or complexity in the level of detail required for the application of these new CFs and the information currently available within companies and supply chains.

5 Conclusions and outlook

This paper is one of the first LCA case studies to attempt to address land use impacts in a more comprehensive way by including a number of new impact categories covering the differentiation of land use types and bio-geographical aspects. The additional value of applying the impact assessment models proposed in this special issue and applied in the margarine case study is not proven, apart from in the

area of bio-geographical differentiation. More research and case studies are needed to ascertain which of the new impact categories are important and to understand the overlap between them. In terms of land use type differentiation, a clear gap has been identified by the use of broad classes of agricultural systems (e.g. annual crops are not distinguished from permanent crops) and further differentiation at least at the second classification level identified by Koellner et al. (2012b) is probably required for those land use types that dominate land occupation (agriculture, including annual and permanent crops, and pastures; forestry for wooden-based products). Differentiation of intensity (e.g. conventional vs. extensive agriculture) in the CF would also be required if those types of agriculture are included in the study; otherwise, the system using less land will always show smaller impacts. In terms of bio-geographical differentiation, the biome level classification provides good insights into potential hotspots, although large uncertainty is probably hidden in the use of single CF for whole biomes. However, obtaining inventory spatial information at a level that is more specific than the biome is likely to be very challenging for many product systems.

One key learning from this study is that occupation seems to be the key driver for land use impacts, more than transformation. Here, yield (as main driver for occupation) tends to dominate the results; as yield is associated with large variability, it will be a source of variability/uncertainty for land use impacts too. It is thus very important to keep transparency at the LCI level in terms of what yields are considered for each part of the product system, and also on the methods and assumptions considered to assess land transformation (LUC).

It is commonly accepted that LCIA provides information on potential, rather than actual, impacts. However, the conceptual approach followed to assess land use impacts relates to a theoretical potential reference and therefore it tends to focus the impact assessment away from the obvious sources of actual impacts such as land use change. In some decision contexts, it may be helpful to explore the use of alternative ecological reference points such as a current reference state (option 3 suggested in Koellner et al. (2012a)) or the alternative most probable land use.

Even though land use impacts have been discussed in the LCA literature for some years, their consistent and practical consideration in case studies is in its infancy. More applications are thus needed that test the limits and potential flaws of the methods and theoretical approaches used in order to gain confidence in the helpfulness of the results as support for decision making.

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